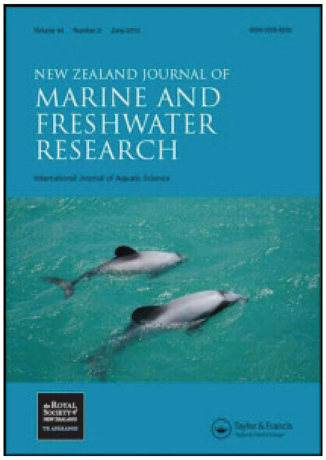

New Zealand Journal of Marine and Freshwater Research
Publication details, including instructions for authors and subscription information:
http://www.informaworld.com/smpp/title~content=t918959567

Environmental effects of sediment on New Zealand streams: A review

Paddy A. Ryan[a]
[a] West Coast Regional Council, Greymouth, New Zealand

Online publication date: 30 March 2010

To cite this Article Ryan, Paddy A.(1991) 'Environmental effects of sediment on New Zealand streams: A review', New Zealand Journal of Marine and Freshwater Research, 25: 2, 207 — 221

To link to this Article: DOI: 10.1080/00288330.1991.9516472

URL: http://dx.doi.org/10.1080/00288330.1991.9516472

New Zealand Journal of Marine and Freshwater Research, 1991, Vol. 25: 207–221
0028–8330/2502–0207 $2.50/0

Environmental effects of sediment on New Zealand streams: a review

PADDY A. RYAN
West Coast Regional Council
P. O. Box 66, Greymouth, New Zealand

Abstract Literature pertaining to sediment in stream ecosystems is reviewed. Suspended sediment can alter the water chemistry, and cause temperature decreases and turbidity increases. Deposition of sediment may change the character of the substrate, block interstices, and reduce interstitial volume. Turbidity levels as low as 5 NTU can decrease primary productivity by 3–13%. An increase of suspended sediment levels increases the drift fauna and may reduce benthic densities as well as alter community structure. Fish are not so obviously affected, although death resulting from clogging of the gills may occur in sensitive species. Suspended and deposited sediment may alter fish community composition, both by interference with run–riffle–pool sequences and by favouring olfactory feeders over visual feeders. In many situations aesthetic reactions to suspended sediment may be of more concern than biological ones. In already turbid water, a 20–50% reduction in clarity may not be detectable whereas in normally clear water a clarity reduction of 10–15% is distinguishable. Recovery from the effects of suspended sediment and sediment deposition is usually rapid, once the source of contamination is removed and as long as the stream is prone to regular spates; the aesthetic recovery may only take days whereas biological recovery may take months.

Keywords suspended sediment; suspended solids; turbidity; NTU; suspensoid; silt; benthos; placer mining; water quality standards; drift; streams; rivers; fish

M90063
Received 23 November 1990; accepted 22 March 1991

INTRODUCTION

Anthropogenic sediment in freshwaters is a by-product of several activities. In New Zealand the major contributor is probably agriculture, closely followed by construction projects. Extractive industries (in particular forestry, coal-mining, alluvial mining, hard rock mining, gravel extraction, china clay mining) and various other activities contribute to a variable extent depending upon the control methods used. The problem is universal and the literature on the subject is truly international. Very few New Zealand studies on the effects of sediment on stream ecosystems have been published to date; none on the effects of sediment generated by alluvial mining. The human organism has also been ignored and the aesthetic effects of suspended sediment have only recently been investigated. This literature review examines the effects of sediments on freshwater ecosystems and gives the results of some of the more important overseas work and relevant New Zealand studies. It should be stressed that because the literature is so voluminous (over 3000 references to suspended sediment were picked up during a computer search) this can be no more than an overview. However, few references are directly relevant to the effects of suspended sediment on stream biota. In 1977, Sorensen et al. stated (with regard to the American situation):

> "Although some 185 journal articles, government reports, and other references were cited herein (about 45% published since 1974) and many other reports (about 300 citations) were reviewed, there is a dearth of quantitative information on the response of freshwater biota, especially at the community level, to suspended and dissolved solids."

The situation has not changed markedly in the intervening 14 years. It should also be stressed that overseas conditions need not be the same as those pertaining to New Zealand. Extrapolation to the local situation is not a substitute for local research.

Effects of sediment on chemical and physical stream characteristics

All streams carry some suspended solids under natural conditions and Brown (1960, in Hellawell, 1986) gave values of up to 300 000 g m^{-3} in extreme floods. Direct chemical effects on streams are minimal. Ellis (1936, in Cordone & Kelly 1961) found that eroded silt did not materially alter the salt complex or amount of electrolytes in river waters. Sediment from overburden removal can also be associated with chemical changes. Dick et al. (1986) showed that surface coal mining in a limestone-dominated watershed produced sediments that contained a higher concentration of calcium, magnesium, and strontium, and had a higher pH than sediment from a sandstone-shale-dominated watershed. Many of the changes associated with mining were reversed when landscape reclamation occurred. They also reported that settling pond effluents contained higher concentrations of most chemical parameters than influent waters. This was attributed to the greater proportion of chemically active fine particles in the pond effluent as a result of the settling of coarser particles.

Where sediment has a high organic content (most notably in run-off from forestry operations), the organic particles undergo anaerobic breakdown when the sediment settles. These breakdown products use dissolved oxygen during further decomposition. Under low-flow conditions (often in summer, when stream oxygen levels may already be low) this can produce a critical oxygen shortage leading to fish kills (at this time oxidised metals may be reduced and feed into the water). For example, Graynoth (1979) described a fish kill in the Motueka River which may have been attributable to forestry operations in the Golden Downs State Forest, Nelson. Clear-felling in north Westland increased catchment sediment yields by up to 100-fold (O'Loughlin & Pearce 1976; O'Loughlin et al. 1980, 1982); erosion from logging roads and loss of soil from tree roots were the prime sediment sources. Other relevant studies include a review of the possible effects of forestry on inland waters of Tasmania (Michaelis 1984), Campbell & Doeg's (1989) review of timber harvesting and production on streams, and Winterbourn's (1986) discussion of forestry practices and stream communities, with reference to New Zealand.

Ryder (1989), in a study of the effects of sediment deposition in a central Otago stream, showed that retention of benthic organic matter is influenced by stone size and interstitial fine-sediment deposition. Although not discussed by Ryder, it seems likely that reduction of water circulation through the subsurface gravel will lead to reduced oxygenation there and a reduction in the carrying capacity of the stream. In Alaskan streams subject to alluvial mining, Bjerklie & LaPerriere (1985) found that the relative water table below the streams fell by an average of around 0.3 m (resulting from a build-up in the height of the stream bed) and dissolved oxygen was consequently reduced in the interflow.

As might be expected, high concentrations of suspended sediment cause problems for any form of water impoundment. As flow velocity diminishes, deposition increases. In the case of hydro dams, this can be a major problem. One solution is dredging, or in some instances, flushing. Flushing dams has unwanted effects on downstream values. If the dam is deep, water in the lower levels is frequently deoxygenated. Flushing the dam releases this often cold and highly turbid water into the receiving stream and can bring about fish kills. An adequate management regime to reduce these problems is discussed by Hesse & Newcomb (1982).

Stream bed sediment deposition, although an inevitable result of suspended sediment transport, is not necessarily the most important effect. The major effect of increased turbidity is probably on the photosynthetic activity of plants. Photosynthetic production will be reduced by turbidity if light is a limiting factor. However, effects on visual predators and aesthetics may be equally important. In some instances, turbidity may decrease the water temperature as more heat is reflected: this may affect temperature-sensitive species.

In some shallow lakes (and presumably slow-flowing rivers), wind action may resuspend bottom silt. If, as sometimes occurs, the silt has a high affinity for oxygen, the net result may be deoxygenation of the water column (Bruton 1985).

Kirk (1985) examined the optical properties of suspended sediment and discussed their implications for primary productivity in aquatic ecosystems. Suspended particles (suspensoids) scatter light and may also absorb it. Intuitively, particles that scatter light would be expected to increase the depth to which light penetrates compared with particles that absorb it. However, scattering increases the path length travelled by individual photons, thereby increasing their likelihood of absorption by absorptive particles, dissolved matter, and water itself. Particles may also reflect photons upwards. In Lake Burley Griffin, in Canberra, Australia, this effect reduces the depth at which photosynthesis can occur from 9 m to about 2 m, demonstrating that the phenomenon is of practical importance (Kirk 1985). Presumably a similar light scattering contribution will be evident in New Zealand's rivers.

Effect of suspended sediment on plants

Stream communities vary in the relative importance of their energy inputs. Small forested streams such as those on New Zealand's West Coast of the South Island, contain a highly diverse benthos (Cowie 1983, 1985). Such streams, partially enclosed by the canopy, may obtain a substantial proportion of their energy inputs from outside the stream. These allochthonous energy sources consist mostly of decaying leaves. The energy they contain is made available to the invertebrate community by action of bacteria, fungi, and the larvae of various insects which mechanically break leaves down into smaller pieces (Winterbourn 1987). This situation contrasts with that encountered further down stream. As the waterway becomes larger, it is less shielded by riparian vegetation and thus receives more light. The surface area of the bed (per unit distance) increases whereas the relative importance of allochthonous inputs is correspondingly reduced. Murphy et al. (1981) showed that primary production was higher in open stream sections than in those sections covered by the canopy. Autochthonous energy sources may become more and more important as the stream gets bigger and the relative amount of stream cover is reduced (Winterbourn et al. 1984).

This leads to the possibility that streams relying on allochthonous input as a community energy source may be less affected by high levels of suspended sediment than those which rely on an autochthonous energy source. Lowland streams, dependent upon the photosynthetic activity of green plants as an energy source, may require lower levels of turbidity. To date I have seen no publication which resolves this issue.

There has been considerable research on primary production in streams affected by suspended sediment. Lewis (1973a) investigated the growth of an aquatic moss (*Eurhynchium riparoides*) down stream of a Welsh coal washery effluent. Abrasive damage to leaf surfaces from coal dust occurred after only 3 weeks at a level of 100 g m^{-3}. Although the moss was able to survive coal fines at concentrations as high as 5000 g m^{-3}, spore germination was reduced by 42% at these concentrations. New side shoots on adult plants occurred only at concentrations below 500 g m^{-3} (Lewis 1973b).

Nuttall & Bielby (1973) noted that in British streams where rooted vegetation would otherwise be expected, it was completely absent if the stream was subject to China-clay wastes. Effluent discharge control standards differed markedly from most New Zealand guidelines: mean suspended sediment in one stream was as high as 46 kg m^{-3}. Despite this enormous loading and an almost total lack of primary production, several stream invertebrates (Tubificidae, Naididae, and Chironomidae) were able to survive at increased densities when compared with control streams.

Sediment resulting from alluvial mining and other activities can have a marked effect on photosynthesis. Van Nieuwenhuyse & LaPerriere (1986) investigated mined and unmined streams in subarctic Alaska and measured the primary production in each. Suspended sediment levels of around 200 mg l^{-1} (in this case 170 NTU) caused a 50% reduction in primary production. This reduction was less than expected because the light reaching the bottom was reduced by 75%. They concluded that either species composition changed or the amount of chlorophyll *a* in each photosynthetic cell increased. In heavily mined streams the turbidity reached 1200 NTU and no primary production occurred. It was not known whether this was because of the reduction of light intensity, scouring by the suspended sediment, or poisoning by heavy metals released from the sediments by the mining. Lloyd et al. (1987) suggested that a turbidity of only 5 NTU from alluvial mining can decrease the primary productivity of shallow clear-water streams by about 3–13%. An increase of 25 NTUs may decrease primary production by 13–50% in shallow streams. Primary production in clear streams of depths greater than 0.5 m would be reduced even further.

Effect of sediment on invertebrates

Spates and concomitant increases in both suspended and deposited sediment are a natural occurrence in streams. It follows therefore that stream organisms must be adapted to withstand occasional increases in suspended and benthic sediment levels. Sediment deposition has a variety of effects on benthos: these range from minor interference with feeding (and hence lower productivity) right through to death by smothering. All of these can occur as a result of natural events. Input from human activities, if as infrequent as natural flooding, may be well handled by the stream community. Continuous high-level inputs are another matter. Because of the frequency of human-induced sediment in stream ecosystems, there has been considerable work on this subject, most of it outside New Zealand. The bulk of anthropogenic sediment is diffuse in nature and it is often difficult to separate the effects of sediment from that of other pollutants. Kelly (1988) pointed out that work on the effects of heavy metals and acidity on aquatic ecosystems has occurred at the expense, perhaps, of associated effects such as those of turbidity and suspended solids. As a result, little work has been done on the effects of alluvial mining, an industry which can produce continuous high-level point source inputs of sediment.

Significant exceptions are studies on the effects of placer mining on the invertebrate communities of Alaskan streams. Wagener and LaPerriere (1985) found a decreased density and biomass of invertebrates in mined streams and, as one of the few studies that may be directly comparable with the New Zealand alluvial mining situation, it is worth quoting their conclusions in full:

"1. Placer mining increased turbidity, settleable solids, and non-filterable residues (suspended sediment).

"2. Sedimentation decreased density and biomass of benthic invertebrates.

"3. Increased turbidity was the strongest descriptor of reduced invertebrate density and biomass.

"... for evaluating the effects of placer mining, invertebrate densities were good indicators of water quality. Many diversity and biotic indices have been developed in temperate regions where invertebrate densities and taxonomic richnesses are high; they therefore may not be appropriate for Alaskan subarctic streams. Density, however, which requires no taxonomic identification, varies inversely with mean turbidity, settleable solids, and non-filterable residues. Acarina seems to be the taxon most sensitive to sediment and may be the best 'indicator organism' of placer mining effects on streams."

Weber & Post (1985) demonstrated reduced aquatic invertebrate populations down stream of mined areas in comparisons between mined and unmined sections of the catchment of an Alaskan creek. A review by Lloyd et al. (1987) summarised information on the effects of turbidity. Densities and biomass of benthic invertebrates were significantly higher in unmined Alaskan streams than in mined streams. Suspended sediment may not be the only hazard to stream life from placer mining: LaPerriere et al. (1985) found various heavy metals were associated with alluvial mining effluents. Some of the effects attributed to suspended sediments by Wagener & LaPerriere (1985) may therefore have resulted from the presence of heavy metals.

Suspended sediment can affect the benthos in several ways. As discussed above, increased turbidity can reduce primary productivity in a water body. This immediately reduces the energy available to enter the food chain and stream productivity may decrease. The situation is not quite so clear-cut in streams which rely primarily on allochthonous energy sources. Thus increased suspended sediment need not significantly reduce the food supply available to invertebrates and, particularly if it remains in suspension, human-induced sediment input may have little effect on the biota.

However, if sediment settles on the substrate, its effect can range from minor to catastrophic. Most of the settling pond effluent from West Coast (South Island, New Zealand) goldmining operations consists of fine fractions of 1 µm or less (West Coast Goldminers Association 1988). These particles are highly mobile and remain in suspension for a long time. It is only when there are "accidents" with settling ponds that major quantities of settleable solids are released. Nonetheless, Graham (1990) has shown that fine silt can be entrapped by periphyton even at very low levels, thus reducing the attractiveness of the periphyton to algal grazers. At suspended mineral silt levels between 1 and 10 g m^{-3}, silt accumulation in epilithic periphyton accounted for about 50% of its dry weight. This, in turn, caused a reduction in the mean organic content of the periphyton to 22% of the dry weight, compared with 52% in a reference stream with suspended sediment levels below 1.0 g m^{-3} during normal flow conditions.

Organisms respond to increased suspensoids in a variety of ways. If the level is high, such as in a flood, the immediate response of some organisms is to move (if they can). The actual displacement depends upon a variety of factors including the species involved and the velocity of the current. Organismal drift is usually maximal at night and minimal during the day (Fowles 1972; Graesser 1988).

Increased levels of turbidity increase the drift, apparently without an increase in the compensatory upstream movements. Luedtke & Brusven (1976) showed that sand deposition prevents upstream movement, presumably because the substrate lacks stability.

Rosenberg & Wiens (1975) demonstrated that increased suspended sediment levels were associated with increased drift rates. They could not correlate the actual response to the sediment level, suggesting that some threshold value may be involved. Ciborowksi et al. (1977) showed that both drift numbers and drift density increased with turbidity during darkness, but not during the day. The lack of response during the day could have resulted from inappropriate experimental design (they used an artificial plexiglass stream) or simply from the fact that fewer animals are active during the day. Gammon (1970) demonstrated that increases of 40 g m^{-3} and 80 g m^{-3} of suspensoids above background levels caused an increase in drift of 25% and 90%, respectively. Graesser (1988), in a study of three flood-prone, South Westland streams showed that numbers in the drift increased with increasing discharge. However an inverse relationship existed between drift density and discharge. The effect of suspended sediment on the drift was not investigated. LaPerriere (1983) considered invertebrate drift to be somewhat analogous to sediment transport in streams. She showed that the inverse relationship between drift density and discharge resulted from a dilution and hypothesised that the active component of drift did not involve the behaviour of swimming up into the water column but rather a behaviour of swimming down to the substrate after being caught up by the water's turbulence.

Ryder (1989) showed a sudden increase in the drift densities of stream insects when sediment was artificially introduced to a natural stream. There was also a reduction in the abundance of some benthic animals. Thus levels of turbidity, too low to cause immediate harm to an organism, may increase the drift rate and lead to a reduction in the density of stream benthos. In the normal course of events there would be a compensating drift from up stream. However, instead of reattaching to the substrate, the increase in turbidity may cause the drift fauna to continue to drift.

Suspensoids also have more obvious effects. When they settle they can interfere with the feeding of benthos by covering the food supply of those organisms that feed on periphyton. In New Zealand, Graham (1990) showed that even at low concentrations of fine silt and clay, epilithic periphyton in riffle habitats may accumulate the suspensoids. Ryder (1989) noted that larvae of *Deleatidium* spp. (Ephemeroptera) and *Pycnocentrodes* sp. (Trichoptera) preferentially grazed on unsilted periphyton rather than on silted periphyton. In addition the growth of early instars of *Pycnocentrodes* fed on silted periphyton was significantly less than of larvae that grazed on unsilted periphyton. Suspended sediments can also block up the nets of net-feeding species (Gammon 1970) and Eddlemon & Tolbert (1983, in Kelly 1988) reported that iron flocs, produced by acid mine drainage, could cause physical abrasion of the benthos.

Other profound effects relate to the clogging of the interstices in the bottom gravel. In a Canadian stream, Pugsley & Hynes (1983) found that 70% of stream insects lived in the top 10 cm of the stream bed. Some live there all the time, others use it for shelter during the day and move onto the tops of stones at night to feed. When the interstices are blocked, interchange of water and metabolites with surface waters is reduced and the interstitial layer may become oxygen-depleted. Although some animals may be capable of forcing their way through this barrier, many presumably are not and die from a lack of oxygen.

In most instances of heavy silt deposition there is a change in community structure rather than a total loss of the fauna. This phenomenon has been well documented in the literature. Soroka & MacKenzie-Grieve (1983) found that numbers of Plecoptera and Ephemeroptera were reduced down stream of Canadian placer mining. Dipterans appeared more tolerant than the other two orders. In addition Hellawell (1986), in a review of physical disturbances on stream communities, summarised the effects as follows:

> "Certain recurrent features will have been noted with respect to the effects of deposition of solids on the benthos. These include the lowering of benthic community diversity through the disappearance or marked reduction in biomass and numbers of certain sensitive species, often those requiring 'open' eroding substrata for attachment or feeding (especially filter feeders). The replacement fauna consists of burrowing forms, typical of soft, depositing substrates, provided that the deposits are neither excessive in their rates of accumulation nor completely sterile and devoid of nutriment. Almost invariably the dominant organisms are chironomid larvae and oligochaete worms and whenever the deposits are also organically enriched then these groups may become extremely abundant."

The total productivity of the changed community can be higher than the original if there is a suitable energy source associated with the sediment (such as from logging activity). This does not mean that the new community composition is acceptable; for instance, some salmonids rely heavily on the drift fauna for their food. If the drift fauna is reduced (as may occur with a change to burrowing forms), the stream becomes a less favourable habitat for salmonids—with a corresponding effect on angling.

Culp et al. (1986) compared the responses of macroinvertebrates to fine-sediment deposition with their responses to fine-sediment transportation. In marked contrast with the above findings, they noted only one taxon that showed a numerical decrease in the benthos and an increase in the drift as a result of deposition. Fine suspended sediment transport reduced total benthic densities by more than 50% in 24 h.

Ryder (1989) suggested that reduced invertebrate abundance in response to fine sediment deposition can be explained by the actual reduction of interstitial space. He qualified this observation by pointing out that the data came from introduced substrates and that the substrate was relatively shallow (c. 3 cm), hence the applicability of these findings to the whole of the substrate is problematical. Thus he found significant positive correlations between interstitial volume and the abundance of invertebrate taxa that are normally reduced in number on substrates with fine sediment. In contrast, the larvae of Elmidae which were found in increased numbers on impacted substrates, were negatively correlated with interstitial space. A 12–17% increase in the interstitial fine sediment of a relatively "sediment-free" substrate was associated with a 16–40% decrease in the abundance of total invertebrates and a 27–55% decrease in the abundance of the ephemeropteran *Deleatidium*.

Preliminary results from a study of seven West Coast, South Island streams subjected to alluvial mining, indicate that a relatively small increase in suspended sediment has a substantial effect on the invertebrate fauna (J. Quinn, Water Quality Centre, DSIR, Hamilton pers. comm.). When matched sites up stream and down stream of mining were compared, it was found that none of the downstream sites had more than half the invertebrate densities of the upstream sites. At two of the downstream sites, mean turbidities were increased by only 5–7 NTU. Thus unimpacted streams may be extremely sensitive to anthropogenic increases of suspended sediment.

Cline et al. (1982) examined the effects of road construction on the macroinvertebrates and algae of a high mountain stream in North Colorado. They found a reduction in algal species diversity and a variety of responses by members of the macroinvertebrate community. Where an alteration in the community occurred there was a reduction in density, abundance, and diversity, together with a change in the taxonomic composition. The severity of the response depended on the flow regime and the timing and duration of the impact. The high stream gradient reduced the effects because silt was rapidly removed from the system.

In summary, suspended sediment loadings on streams may affect stream faunas in several ways. High turbidity reduces photosynthesis of plants and thus overall productivity of the community. It also promotes drift and loss of animals from affected areas. High suspensoid levels may clog the food-filtering or trapping apparatus of stream insects. Deposits of sediment can coat stone surfaces, both reducing the available food supply and eliminating attachment points for those animals such as larval simuliids (blackflies) which need to anchor themselves to the substrate. In addition, reduction of interstitial spaces within the stream bed means less habitat and reduced exchange of oxygen and metabolites for animals living in the benthos. In "worst-case" scenarios, entire stream sections can be rendered practically devoid of all macroinvertebrates apart from a few species adapted to living in deoxygenated silt. As mentioned earlier, streams highly dependent upon allochthonous material may be less affected by sediment than those dependent upon autochthonous sources, but this has yet to be demonstrated.

Effects of suspended sediment on fish

The direct effects of suspended sediment on fish are much better documented than for other organisms. There are obvious reasons for this: fish are economically important and provide a large number of people with a pleasant leisure time activity. The rest of the stream community, by way of contrast, offer none of these apparent benefits. Ultimately, the effect of most anthropogenic activity is reflected in the health of the fish population because of direct impacts and/or food chain related effects. The most complete volume of information on the subject is by Alabaster & Lloyd (1982).

As is the case with the effects of sediment on plants and stream communities, little work has been done on New Zealand fish. Although it may be safe to apply the results of overseas studies to trout and salmon in New Zealand, it is not wise to do so with regard to our native fishes.

Graynoth (1979) examined the consequences of logging on stream environments in Nelson. He discovered that the number of dwarf galaxias (*Galaxias divergens*) and longfin eels (*Anguilla dieffenbachii*) decreased as a result of logging activities. It was not clear whether this was a result of sediment concentration, lack of cover, stream temperature, or a combination of all three.

Outside New Zealand, most notably in the United States, the effects of suspended sediment on fish has been the focus of much attention. This will not appear surprising as excessive siltation from anthropogenic erosion occurs in 34% of all American streams and was considered the most important factor limiting usable fish habitat (Judy et al. 1984 in Lloyd et al. 1987).

Berkman & Rabeni (1987) evaluated the effects of siltation on stream fish in north-east Missouri. As the percentage of fine substrate increased, the distinction between riffle, run, and pool communities decreased. This led to a reduction in fish diversity. Although the New Zealand native fish fauna is not as specialised or as diverse as that in Missouri, a reduction in riffle areas would reduce the suitability of the habitat for trout and a number of native fish, e.g., the blue-gilled bully (*Gobiomorphus hubbsi*), the upland bully (*Gobiomorphus breviceps*), the torrentfish (*Cheimarrichthys fosteri*), and several galaxiids (McDowall et al. 1977). This, of course, presupposes long-term sediment pollution. Gradall & Swenson (1982) demonstrated that increasing turbidity attracted chub (*Semotilus atromaculatus*) but had no apparent effect on brook trout (*Salvelinus fontinalis*). The implication is that high turbidities can favour one fish species over another and thus alter community composition.

Other North American studies have concentrated more on the effects of suspensoids on the behaviour of individual fish species rather than on communities. This information is useful, but is difficult to extrapolate to freshwater ecosystems, particularly when species composition is different. Juveniles of coho salmon (*Oncorhynchus kisutch*), a species not found in New Zealand, avoid suspended sediment levels higher than 70 NTU. Those acclimatised to levels normally found in pristine streams (around 2–15 NTU) did not exhibit an avoidance reaction until the level reached 100 NTU (Bisson & Bilby 1982). Sigler et al. (1984) conducted laboratory tests on the growth of rainbow trout (*Oncorhynchus mykiss*) and coho salmon. When exposed to turbidities of 100–300 NTU, fish either left the experimental channels or died, but in the 25–50 NTU range they didn't leave—although in most cases their growth was significantly lower than that of fish in clear water. Reynolds et al. (1989) demonstrated that sac fry of arctic grayling *Thymallus arcticus* were vulnerable to elevated suspended sediment levels. Water containing 400 g m^{-3} caused 50% mortality of caged sac fry whereas fingerling and juvenile fish survived short term exposure (1–9 days) to high concentrations of mining sediments (>3000 g m^{-3}). Similarly, O'Scannell (1988) found no effect when arctic grayling fingerlings were exposed to a turbidity of 445 NTU for 96 h. However, in a laboratory choice chamber, grayling avoided water with a turbidity above 20 NTU and their reactive distance to prey diminished in proportion to the natural logarithm of turbidity. O'Scannell makes the important point that reactive distance is proportional to reactive volume. A reduction of 50% in reactive distance corresponds to a 90% reduction in reactive volume. When this is combined with a decline in available prey items, it is evident that visual feeders may be badly affected by suspended sediment.

Further evidence that turbid water may interfere with feeding success is presented by Gardner (1981). He fed *Daphnia* to bluegills (*Lepomis macrochirus*) at various levels of turbidity. 60 NTU produced a 20% reduction in feeding success compared with "clear water" controls. Breitburg (1988) showed that striped bass (*Morone saxatilis*) larvae consumed c. 40% fewer prey in suspensoid concentrations of 200 and 500 g m^{-3} than in 0 or 75 g m^{-3}, respectively. In contrast, striped bass larvae feeding on *Daphnia pulex* captured the same average number of prey at all suspensoid concentrations tested. Turbidity had no effect on the size of copepods or *D. pulex* eaten. Surprisingly perhaps, estuarine larval Pacific herring (*Clupea harengus*) fed at maximum intensity at elevated turbidity levels (Boehlert & Morgan 1985). Even at 1000 g m^{-3}, the young fish fed better than in clear water: on the small perceptive scale used by young fish, visual contrast of prey items was enhanced. Contrary to expectation, this study showed that the dirty water was making it easier for the fish to find food.

Higher levels of suspended sediment may have more profound effects on fish. Benthic food organisms may be smothered by silt at levels well below those having a directly harmful effect on fish. Under these circumstances the impact is twofold. Not only are the food items reduced in number but they are also harder for visually-feeding fish to locate.

Silt deposition may also cause widespread egg mortality and many salmonids will not spawn in gravels that have become silted (Alabaster & Lloyd 1982). Clavel et al. (in Rivier & Seguier 1985) stated that in the River Allier, exposure of brown trout (*Salmo trutta*) eggs to suspended sediment concentrations of between 20 and 100 g m^{-3} for 20 days resulted in 75% mortality of the eggs (as compared with 20% in the controls). Koehn & O'Connor (1990) suggested that in silted Australian streams the freshwater blackfish (*Gadopsis marmoratus*) may not be able to find the clean sites required for egg adhesion. Blyth & Jackson (1985) suggested that the eggs and larvae may also be affected by sediment deposition.

At yet higher levels, the silt may have direct lethal effects on fish by clogging gillrakers and gill filaments (Bruton 1985). Acceptable levels may depend upon the nature of the silt, the concentration of the silt, the degree of oxygenation of the water, water temperature, the species of fish, the size of the fish and the concentration of suspended sediment to which it has become acclimated. In streams that are naturally "dirty", resident fish species can be expected to tolerate higher suspensoid levels than will the same species in naturally "clean" streams.

Aesthetic effects of suspended sediment on streams

The acceptable level of suspended sediment depends to a large extent on the vociferousness of the pressure group involved. Lloyd et al. (1987) stated that it is generally acknowledged that turbid water is less acceptable than clear water for consumption, contact recreation, and aesthetic enjoyment. In much of New Zealand, where clear freshwater is not in short supply, aesthetic considerations may over-ride ecological ones. Lloyd et al. note that turbidities of 8–50 NTU, 40 km down stream from mine discharges on the "naturally clear" Chatanika River, coincided with, and may have contributed to, a 55% decline in sport fishing. The Chatanika River, once the second most popular water body for sport fishing in Alaska, fell to seventh when mining began. Similar situations have been reported in other countries (Lloyd et al. 1987).

Turbidity on its own is not a complete measure of the aesthetic effects of suspensoids. The problem arises from the perceived clarity of the water as depth changes. A turbidity of 10 NTU may look acceptable when the stream is only a few centimetres deep. However it may look quite unacceptable when the stream is 1 m or deeper. Similarly, 10 NTU in a fast-flowing stream where white water and bubbles ameliorate the visual effects of the sediment, may look more acceptable than 10 NTU in a slow-flowing stream.

Increased sediment levels are also usually more acceptable in rivers and streams that are naturally dirty for much of the year, e.g., Grey and Hokitika Rivers. A small increase in suspended sediment in a naturally clear stream is immediately obvious. A large increase in a naturally dirty stream may not be.

Kirk (1988) suggested that water managers should specify that effluent must not increase the vertical attenuation coefficient of photosynthetically available radiation (PAR) by more than x%, and the reflectance of PAR by more than y%, or decrease the Secchi depth by more than z%. He did not offer values for x, y, and z, but remarked that they should certainly be low.

Davies-Colley (1988a) has addressed the question of "acceptable" reduction in water clarity. He suggested that a 20–50% reduction in clarity may be the detectable threshold for the human eye when the colour induced by suspended sediment is the only cue.

Perceptions of what is acceptable in a water body will depend upon the use to which it may be put. Smith et al. (in press) investigated the water clarity criteria for bathing waters, based upon user perception. They found that bathing water quality assessment was strongly related to visual cues, in particular water clarity. A minimum water clarity (as measured by horizontal black disc sighting range y_{BD}, Davies-Colley 1988b) of 1.2 m (which corresponds to a Secchi disc depth of 1.5 m) is required before a water is perceived, on the average, as suitable for bathing. If, for management purposes, it is required that 90% of people perceive a water clarity as suitable for bathing, then y_{BD} needs to exceed about 2.2 m, corresponding to a Secchi disc depth of c. 2.75 m. The National Technical Advisory Committee to the Secretary of the Interior, Federal Water Pollution Control Administration, Washington D.C., recommended (1968) that a Secchi disc should be visible at a depth of 1.2 m. This value has subsequently been included in several water quality compilations (Hart 1974; Canadian Water Quality Guidelines 1987). However, at a Secchi depth of 1.2 m, the substrate (being of lower contrast than a Secchi disc) in 1.2 m deep water is not visible (Smith, Water Quality Centre, Hamilton pers. comm.). A Secchi depth of about 2 m may be required before the bottom (depending on its apparent contrast) can be seen by a bather (e.g., Davies-Colley & Smith 1990). The Smith et al. study (in press) appears to be the first to attempt to quantify what the public actually require in a bathing water situation.

Davies-Colley & Smith (1990) injected a highly scattering calcite slurry into a small stream and used a panel of school students to record their perceptions of clarity changes. In this case, reductions of clarity of 10–15% were detected. Perceived changes of clarity in already turbid water may be closer to the 20–50% reduction in clarity suggested by Davies-Colley in his earlier study. In situations where aesthetic considerations are important, clarity reductions of 20–50% are a starting point for setting standards.

In general, the public at large will be upset by what they perceive as any degradation of their recreational resource. Applied standards may be driven by public perceptions rather than by scientific evidence but the Smith et al. study shows that the two can be reconciled.

Recovery from high sediment deposition

The discussion to date has centred upon the effects of turbidity and suspended sediment in water ways. However, it is important in all resource-based activities to distinguish between permanent, long-term, medium-term, and short-term effects. In most societies the shorter the effect, the less objectionable it is. This is of particular concern on the West Coast of the South Island of New Zealand where the effects of alluvial mining have attracted considerable adverse comment. If alluvial mining had long-term harmful effects, much opposition would be justified and reasonable. Overseas evidence and local experience suggest that with adequate stream rehabilitation, recovery is rapid. Hamilton (1961) showed that severe and persistent sedimentation is required to induce faunal changes. Hill (1972, in Sorensen et al. 1977) found that sediment caused by strip mining was responsible for limiting populations of benthos and fish. Complete reclamation of spoil areas reduced levels of siltation and turbidity, and allowed recovery of the community. Luedtke et al. (1976) suggested that complete stream rehabilitation depends upon two factors: elimination of the sediment source and the ability of the stream to flush out the deposited material. Tsui & McCart (1981) determined that pipeline construction in British Columbia which produced high levels of stream sediment had only short-term, non-residual effects on macroinvertebrates. Cline et al. (1982) investigated the effects of road construction on a high mountain stream in Colorado. They found rapid stream community recovery (within a year) from the construction activities and concluded that high snowmelt discharge removes fine sediments which are not subsequently redeposited. Similar findings were reported by Barton (1977). Soroka & McKenzie-Grieve (1983), in a study of placer mining in Canada, found a rapid recovery of the stream bed after periods of higher flow. Accumulation of sediment occurred during low flows. Hellawell (1986) believed that a return to a normal stream community, after domination by silt-loving organisms, would occur after winter spates. Thomas (1985) examined the effects of small suction gold dredges on a Montana stream. Mean insect abundance decreased greatly in the dredged area but recovered rapidly and recolonisation was essentially complete after a month.

Several New Zealand studies have examined the recolonisation of various substrates which have not been impacted with sediment. Sagar (1983) investigated invertebrate recolonisation of introduced substrates of dry channels in a braided river in Canterbury. Spates increased both abundance and diversity of the substrates. He concluded in this, and a later study (Sagar 1986), that unstable river systems are characterised by rapid colonisation mechanisms and low species diversity. Scrimgeour et al. (1988) and Scrimgeour & Winterbourn (1989) found similarly rapid recovery. Eldon et al. (1989) showed that a stream diversion reach on the West Coast of New Zealand's South Island was colonised by a variety of fish species. Pre-diversion levels were achieved within a year by brown trout (*Salmo trutta*) and by the native blue-gilled bully (*Gobiomorphus hubbsi*), and the invertebrate densities and species composition were restored within 91 days.

There are few New Zealand studies documenting recolonisation after sediment deposition. Winterbourn & Rounick (1985) showed that streams on the West Coast of the South Island suffered a severe short-term impact from logging activity-derived, deposited sediment. However, subsequent storms removed almost all of these deposits; insect-dominated communities similar to those present before logging rapidly re-established (Winterbourn 1986). Ryder (1989) concluded, from a series of recolonisation experiments, that flushing of stream beds containing fine interstitial sediments will be followed by rapid invertebrate recolonisation. Those organisms which are prominent in the drift will produce greater benthic densities than those which are not.

Not all studies have shown a rapid return to pre-sediment deposition conditions. Platts et al. (1989) noted that even 20 years after logging around the South Fork River, Idaho, the fine-sediment stores were still decreasing. In New Zealand, Beschta (1983, in Doeg & Koehn 1990) noted that a 150-year return storm delivered sediment to a stream and coarse sediment was still detectable 30 years later. Beschta & Jackson (1979) suggested that scouring of sediment by high flows may only penetrate to a depth of a few centimetres. Carling (1988), working in the United Kingdom, concluded that for within-channel flow there is a threshold which separates flows which winnow fine material from those which entrain the finer bed gravels. This threshold occurred at 60% bankfull. Few, if any, data on bankfull levels are available for New Zealand rivers. Furthermore the steep gradients of West Coast, South Island streams and the braided nature of many East Coast rivers make the application of the 60% bankfull threshold for winnowing suspended sediment difficult.

Standards outside New Zealand

Standards vary from country to country. In general those countries with already degraded waters set levels that would be considered too high for countries in which the resource is still relatively unsullied. The best known standards are those suggested by the European Inland Fisheries Advisory Commission (EIFAC) and reproduced by Alabaster & Lloyd (1982). These standards suggest that there is no evidence that concentrations of suspended solids less than 25 g m^{-3} have harmful effects on fisheries. Alabaster & Lloyd (1982) also stated that it **should usually be possible** to maintain good or moderate fisheries in waters which normally contain 25–80 g m^{-3} suspended solids (**my emphasis**). It should be pointed out that EIFAC (1965) were concerned with all fisheries, including coarse fish and eels, not just salmonids. They were also dealing with waters that had already been degraded to a very large extent. Restoration no longer being possible, these standards constituted a form of "damage control". It should also be mentioned that what constitutes a good fishery in Europe is entirely different from what constitutes a good fishery in New Zealand: expectations in relatively unspoilt New Zealand are considerably higher.

In the United States, standards vary from state to state. A range of these standards is reproduced in Table 1. Lloyd (1987) recommended 5 NTU above natural conditions in salmonid streams and lakes for a high level of protection. He added that extremely clear water bodies may require an even stricter standard because of the dramatic initial impact of turbidity on light penetration (Lloyd et al. 1987). Current Yukon standards (Yukon Fisheries Protection Authorization 1988) allow no increase in sediment in salmonid spawning streams but up to 200 g m^{-3} for rearing streams. In view of the evidence already presented about the effect of suspended sediment on salmonids (e.g., Bisson & Bilby 1982; Sigler et al. 1984), these levels are unacceptably high.

To protect aquatic life, Canadian Water Quality Guidelines (1987) recommended that suspended sediments should not be increased by more than 10 mg l^{-1} where the background level is up to 100 mg l^{-1}. Where the background level is greater than 100 mg l^{-1} the increase should be no more than 10%.

Table 1 Numerical turbidity standards for protection of fish and wildlife aquatic habitats in Alaska and other states (from Lloyd 1987).

State	Turbidity (NTU or JTU)[a]
Alaska	25 units above natural in streams 5 units above natural in lakes
California	20% above natural, not to exceed 10 units above natural
Idaho	5 units above natural
Minnesota	10 units
Montana	10 units (5 above natural)[b]
Oregon	10% above natural
Vermont	10 units (cold water)
Washington	25 units above natural (5 and 10 above natural)[c]
Wyoming	10 units above natural

[a]Nephelometric (NTU) and Jackson (JTU) turbidity units are roughly equivalent

[b]Montana places the more stringent limit on waters containing salmonid fishes

[c]Washington has different values for "excellent" and "good" classes of water

Standards employed by the West Coast Regional Council

The West Coast Regional Council currently allows a mixing zone of 200 m in small streams. Most recent discharge standards imposed on alluvial miners (our major point source of sediment) allow an increase of 10 NTU or 10 g m^{-3} of suspended solids in small streams. In the Grey River (median flow 241 $m^3 s^{-1}$) discharge standards for a major gold dredge operation allowed a turbidity increase up to 3.2 NTU (8 km down stream of the discharge) when ambient turbidity was below 2 NTU, and an increase of 60% when ambient turbidity was 2–16 NTU. When ambient turbidity was above 16 NTU, an increase of 10 NTU was sanctioned. The figures were arrived at after extensive consultation with various interest groups. This sliding scale recognises that under turbid conditions a dirtier discharge is acceptable. In Nelson's Creek, a popular summer picnic and swimming spot, the Council allowed a 2 NTU increase over ambient during "summer" (November to March inclusive) and 10 NTU during the rest of the year.

The question of multiple discharges is being addressed but current thinking suggests that in pristine rivers a maximum acceptable percentage increase over ambient will be determined. Users will be given a proportion of this on a "first-come first-serve basis". Once the dilution capacity of the water way has been reached, additional polluters will only be accepted if they negotiate for a share of an existing users allowance. Ambient will be defined as immediately above the most upstream user.

Streams are most vulnerable to anthropogenic sediment under low-flow conditions. At higher ambient levels of suspended sediment, anthropogenic input becomes proportionately less important. Thus, under flood conditions, human input is of negligible significance.

Water management is full of compromises and on the west coast of New Zealand's South Island clear water is not in short supply. Standards which would totally protect aesthetic and biological values are probably too rigorous for most point source polluters to meet and for water managers to detect. The choice is between pristine waters and no industry, or some development and slight short-term degradation of water quality.

CONCLUSIONS

The effects of suspended sediment on stream ecosystems are many and varied. It could be argued that any increase in sediment carried by a stream could have a detrimental effect on stream ecosystems as well as aesthetic values. This is probably true of sediments introduced by both natural and anthropogenic processes. Streams have a differing capacity to cope with suspended sediment depending upon (amongst other factors) their fauna, their gradient, and the nature of the sediment. In many instances, especially at the lower end of any proposed standards, aesthetic considerations could well over-ride ecological ones. In many montane streams, photosynthetic activity may make a minority contribution to the food supply. If this is so, a high turbidity level may be less damaging than would otherwise be the case. In clear lowland streams, inputs of only 5 NTU above ambient can measurably reduce photosynthesis and hence the productivity of the community.

If there is sediment deposition, reduction in stream productivity is likely to be greater than if the sediment remains in suspension. However, many New Zealand mining settling pond effluents contain very mobile fine fractions (<10 μm) and hence settling is usually slow. Deeper rivers will be more affected than shallow ones. Similarly, as discussed previously, aesthetic considerations are important. Deeper rivers (other factors being equal) can dilute greater quantities of suspensoids and so tighter standards should be possible.

A 10% increase over ambient would be almost undetectable under low-flow conditions (Davies-Colley & Smith 1990) and should protect biological as well as aesthetic values. However, a 10% increase in streams with turbidities up to 5 NTU is essentially beyond detection limits. For a substantial degree of both aesthetic and environmental protection, an increase of 1 NTU (1 g m^{-3}) at ambient levels of up to 10 NTU (10 g m^{-3}) and 10% of ambient thereafter is recommended.

In areas where strict controls on suspended sediment are not considered necessary, less rigid standards could be applied. In small streams, an increase of 5 NTU above ambient will give a measure of environmental protection. For large rivers, 1 or 2 NTU above ambient may be reasonable. In streams with steep gradients where recovery from sediment deposition is likely to be swift, a level of 25 NTU above ambient could be acceptable. However it must be remembered that there will be an effect on receiving waters down stream. If these are recreationally sensitive, then upstream standards must be modified accordingly. This is particularly true of streams emptying into lakes. Rigorous standards must be applied in the catchments of clear-water lakes.

The ideal standard would be a sliding scale which allows small increases of turbidity or suspended sediment under low-flow (clear) conditions and allows proportionately more anthropogenic sediment as the natural level increases. However, as flow rate and ambient suspended sediment and turbidity levels are positively correlated, there is a considerably greater dilution factor available to a point source polluter under flood conditions so a fixed percentage increase over ambient may prove adequate.

In streams impacted by sediment from industrial or agricultural sources, recovery will be rapid once the input of sediments ceases and if sediment scouring occurs during freshes.

Finally, it must be acknowledged that decisions regarding what constitutes acceptable levels of anthropogenic sediment in stream environments are essentially political. Even unmeasurable levels of sediment must produce some effect in a stream. There is thus a continuum of effect from minor to profound. Water managers can only recommend an arbitrary point along this continuum. There will always be conflicting requirements for water and the levels ultimately arrived at will be compromises, often to no one's complete satisfaction.

ACKNOWLEDGMENTS

Thanks to the Westland Catchment Board and Regional Water Board and the West Coast Regional Council for funding this study. Thanks also to the large number of

New Zealand scientists who have so willingly shared their information and made me feel welcome on my return to New Zealand after many years overseas. Special thanks to Gael Ogilvie, Paul Sagar, Greg Ryder, and Chris Hickey for providing obscure references and information. Mike Winterbourn, Rob Davies-Colley, John Quinn, Donald Scott, and David Smith did all of the above as well as comment on the draft manuscript. Thanks also to Marjo Gisberts of SATIS for her efficiency in obtaining hard-to-find references. I am particularly appreciative of the constructive comments of Ian Campbell and Jacqueline LaPerriere.

REFERENCES

Alabaster, J. S.; Lloyd, R. 1982: Water quality criteria for freshwater fish. London, Butterworth. 361 p.

Barton, B. A. 1977: Short-term effects of highway construction on the limnology of a small stream in southern Ontario. *Freshwater biology 7*: 99–108.

Berkman, H. E.; Rabeni, C. F. 1987: Effect of siltation on stream fish communities. *Environmental biology of fishes 18*: 285–294.

Beschta, R. L., Jackson, W. L. 1979. The intrusion of fine sediments into a stable gravel bed. *Journal of the Fisheries Research Board of Canada 36*: 204–210.

Bisson, P. A.; Bilby, R. E. 1982: Avoidance of suspended sediment by juvenile coho salmon. *American journal of fisheries management 4*: 371–374.

Bjerklie, D. M.; LaPerriere, J. D. 1985: Gold-mining effects on stream hydrology and water quality, Circle Quadrangle, Alaska. *Water resources bulletin 21*: 235–243.

Blom, B. E.; Jenkins, T. F.; Leggett, D. C.; Murrmann. R. P. 1976: Effect on sediment organic matter on migration of various chemical constituents during disposal of dredged material. *Dredged material research program. Contract report D–76–7, U.S. Army Engineer Waterways Experiment Station,* Vicksburg, Mississippi. 39180. 182 p.

Blyth, J. D.; Jackson, P. J. 1985. The aquatic habitat and fauna of East Gippsland, Victoria. *Bulletin of the Australian Society for Limnology 10*: 89–109.

Boehlert, G. W.; Morgan, J. B. 1985: Turbidity enhances feeding abilities of larval Pacific herring, *Clupea harengus pallasi. Hydrobiologia 123*: 161–170.

Breitburg, D. L. 1988: Effects of turbidity on prey consumption by striped bass larvae. *Transactions of the American Fisheries Society 117*: 72–77.

Brown, E.; Nishioka, Y. A. 1967: Pesticides in selected western streams. A contribution to the national program. *Journal of environmental quality 4*: 33–40.

Bruton, M. N. 1985: The effects of suspensoids on fish. *Hydrobiologia 125*: 221–241.

Campbell, I. C.; Doeg, T. J. 1989: Impact of timber harvesting and production on streams : a review. *Australian journal of marine and freshwater research 40*: 519–539.

Canadian Water Quality Guidelines 1987: Inland Waters Directorate, Environment Canada, Ottawa, Ontario, Canada.

Carling, P. 1988: The concept of dominant discharge applied to two gravel-bed streams in relation to channel stability thresholds. *Earth surface processes and landforms 13*: 355–367.

Ciborowksi, J. H.; Pointing, P. J.; Corkum, L. D. 1977: The effect of current velocity and sediment on the drift of the mayfly *Ephemerella subvaria* Mcdunnough. *Freshwater biology 7*: 567–572.

Cline, L. D.; Short, R. A.; Ward, J. V. 1982: The influence of highway construction on the macroinvertebrates and epilithic algae of a high mountain stream. *Hydrobiologia 96*: 149–159.

Cordone, A. J.; Kelly, D. W. 1961: The influences of inorganic sediment on the aquatic life of streams. *California fish and game 47*: 189–228.

Cowie, B. 1983: Macroinvertebrate taxa from a southern New Zealand montane stream continuum. *New Zealand entomologist 7*: 439–447.

————1985: An analysis of changes in the invertebrate community along a southern New Zealand montane stream. *Hydrobiologia 120*: 35–46.

Culp, J. M.; Wrona, F. J.; Davies, R. W. 1986: Responses of substrate particle size and detritus to stream benthic macroinvertebrate microdistribution. *Canadian journal of zoology 64*: 1345–1351.

Davies–Colley, R. J. 1988a: Water appearance of the Grey River: assessment of impacts of fine suspended sediments with regard to alluvial gold mining. *Water Quality Centre DSIR Consultancy Report T7068*. Prepared for Grey River Gold Mining Ltd., Ngahere.

————1988b: Measurement of water clarity using a black disk. *Limnology and oceanography 33*: 616–623.

Davies-Colley, R. J.; Smith, D. G. 1990: A panel study on the detectability of change of turbidity of water induced by discharge of inorganic suspensoids to a small stream. *Water Quality Centre publication 17*. Water Quality Centre, Department of Scientific and Industrial Research, Hamilton, New Zealand.

Dick, W. A.; Bonta, J. V.; Haghiri, F. 1986: Chemical quality of suspended sediment from watersheds subjected to surface coal mining. *Journal of environmental quality 15*: 289–293.

Doeg, T. J.; Koehn, J. D. 1990: A review of Australian studies on the effects of forestry practices on aquatic values. Victorian Government, Department of Conservation and Environment, Fisheries Division. Silvicultural Systems Project. *Technical report 5*. 81 p.

Eddlemon, G. K.; Tolbert, V. R. 1983: Chatanooga shale exploitation and the aquatic environment: the critical issues. *Environment international 9*: 85–95.

Eldon, G. A.; Sagar, P. M.; Taylor, M. J.; Bonnett, M. L.; Kelly, G. R. 1989: Recolonisation of a stream diversion by aquatic invertebrates and fish, Big Hohonu River, Westland. *New Zealand freshwater fisheries report 115*, Freshwater Fisheries Centre, MAF Fisheries Christchurch. 53 p.

Ellis, M. M. 1936: Erosion silt as a factor in aquatic environments. *Ecology 17*: 29–42.

European Inland Fisheries Advisory Commission (EIFAC) 1965: Working party on water quality criteria for European freshwater fish. Report on finely divided solids and inland fisheries (EIFAC technical paper 1), *Air and water pollution 9*: 151–168.

Fowles, C. R. 1972: The invertebrate fauna of the South Branch of the Waimakiriri River, Canterbury. Unpublished MSc thesis, University of Canterbury, Christchurch, New Zealand.

Gammon, J. R. 1970: The effect of inorganic sediment on stream biota. *US Environmental Protection Agency, Water pollution control research series* 18050 DWC 12/70. 141 p.

Gardner, M. B. 1981: Effects of turbidity on feeding rates and selectivity of bluegills. *Transactions of the American Fisheries Society 110*: 446–450.

Gradall, K. S.; Swenson, W. A. 1982: Responses of brook trout and creek chubs to turbidity. *Transactions of the American Fisheries Society 111*: 392–395.

Graesser, A. K. 1988: Invertebrate drift in three flood–prone streams in South Westland, New Zealand. *Verhandlungen, internationale Vereinigung für theoretische und angewandte Limnologie 23*: 1422–1426.

Graham, A. A. 1990. Siltation of stone–surface periphyton in rivers by clay–sized particles from low concentrations in suspension. *Hydrobiologia 199*: 107–115.

Graynoth, E. 1979: Effects of logging on stream environments and faunas in Nelson. *New Zealand Journal of marine and freshwater research 13*: 79–109.

Hamilton, J. D., 1961: The effect of sand–pit washings on a stream fauna. *Verhandlungen, internationale Vereinigung für theoretische und angewandte Limnologie 14*: 435–439.

Hart, B. T. 1974. A compilation of Australian water quality criteria. *Australian Water Resources Council technical paper 7*. Canberra, Australian Government Publishing Service.

Hellawell, J. 1986: Effects of suspended solids. *In*: Biological indicators of freshwater pollution and environmental management. London and New York, Elsevier Applied Science Publishers.

Hesse, L. W.; Newcomb, B. A. 1982: Effects of flushing Spencer Hydro on water quality, fish, and insect fauna in the Niobrara River, Nebraska. *North American journal of fisheries management 2*: 45–52.

Judy, R. D. Jr.; Seeley, P. N.; Murray T. M.; Svirsky S. C.; Whitworth, M. R.; Ischinger, L. S. 1984: 1982 National fisheries survey. Volume I. Technical report: initial findings. *United States Fisheries and Wildlife Service*, FWS/OBS–84/06. 140 p.

Kelly, M. 1988: Mining and the freshwater environment. London and New York, Elsevier Applied Science. 231 p.

Kirk, J. T. O. 1985: Effects of suspensoids (turbidity) on penetration of solar radiation in aquatic ecosystems. *Hydrobiologia 125*: 195–208.

————1988: Optical water quality—what does it mean and how should we measure it? *Journal of the water pollution and control federation 60*: 194–197.

Koehn, J.; O'Connor, B. 1990. Threats to Victorian native freshwater fish. *Victorian naturalist 107*: 5–12.

LaPerriere, J. D. 1983: Alkalinity, discharge, average velocity and invertebrate drift concentration in subarctic Alaskan streams. *Journal of freshwater ecology* 2: 141–151.

LaPerriere, J. D.; Wagener, S. M.; Bjerklie, D. M. 1985: Gold-mining effects on heavy metals in streams, Circle Quadrangle, Alaska. *Water resources bulletin 21*: 245–252.

Lewis, K. 1973a: The effect of suspended coal particles on the life forms of the aquatic moss *Eurhynchium riparoides* (Hedw.) I. The gametophyte plant. *Freshwater biology 3*: 251–257.

————1973b: The effect of suspended coal particles on the life forms of the aquatic moss *Eurhynchium riparoides* (Hedw.). II. The effect on spore germination and regeneration of apical tips. *Freshwater biology 3*: 391–395.

Lloyd, D. S. 1987: Turbidity as a water quality standard for salmonid habitats in Alaska. *North American journal of fisheries management* 7: 34– 45.

Lloyd, D. S.; Koenings, J. P.; LaPerriere, J. D. 1987: Effects of turbidity in fresh waters of Alaska. *North American journal of fisheries management* 7: 18–33.

Luedtke, R. J.; Brusven, M. A. 1976: Effects of sand sedimentation on colonization of stream insects. *Journal of the Fisheries Research Board of Canada 33*: 1881–1886.

Luedtke, R. J.; Brusven, M. A.; Watts, F. J. 1976: Benthic insect community changes in relation to in-stream alterations of a sediment-polluted stream. *Melanderia 23*: 21–39.

McDowall, R. M.; Graynoth, E.; Eldon, G. A. 1977: The occurrence and distribution of fishes in streams draining the beech forests of the West Coast and Southland, South Island, New Zealand. *Journal of the Royal Society of New Zealand* 7: 405–424.

Michaelis, F. B. 1984. Possible effects of forestry on inland waters of Tasmania: a review. *Environmental conservation 11*: 331–343.

Murphy, M. L.; Hawkins, C. P.; Anderson, N. H. 1981: Effects of canopy modification and accumulated sediment on stream communities. *Transactions of the American Fisheries Society 110*: 469–478.

National Technical Advisory Committee 1968: *Water Quality Criteria*. Report of the National Technical Advisory Committee to the Secretary of the Interior, Federal Water Pollution Control Administration, Washington D.C., USA.

Nuttall, P. M.; Bielby, G. H. 1973: The effect of China-clay wastes on stream invertebrates. *Environmental pollution 5*: 77–86.

O'Loughlin, C. L.; Pearce, A. J. 1976: Influence of Cenozoic geology on mass movement and sediment yield response to forest removal, North Westland, New Zealand. *Bulletin of the International Association of Engineering Geology 14*: 41–46.

O'Loughlin, C. L.; Rowe, L. K.; Pearce, A. J. 1980: Sediment yield and water quality responses to clearfelling of evergreen mixed forests in western New Zealand. *In*: The influence of man on the hydrological regime with special reference to representative and experimental basins. (*Proceedings of the Helsinki Symposium, June 1980). IAHS publication 130*: 285–292.

O'Loughlin, C. L.; Rowe, L. K.; Pearce, A. J. 1982: Exceptional storm influences on slope erosion and sediment yield in small forest catchments, North Westland, New Zealand. *In*: The First National Symposium on Forest Hydrology, Melbourne. Institution of Engineers, Australia. 8 p.

O'Scannell, P. O. 1988: Effects of elevated sediment levels from placer mining on survival and behavior of immature Arctic grayling. Unpublished MSc thesis. University of Alaska Fairbanks, Fairbanks, Alaska. 93 p.

Platts, W. S., Torquemada, R. J., McHenry, M. L., Graham, C. K. 1989: Changes in spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. *Transactions of the American Fisheries Society 118*: 274–283.

Pugsley, C. W.; Hynes, H. B. N. 1983: A modified freeze-core technique to quantify the depth distribution of fauna in stony streambeds. *Canadian journal of fisheries and aquatic sciences 40*: 637–643.

Reynolds, J. B.; Simmons, R. C.; Burkholder, A. R. 1989: Effects of placer mining discharge on health and food of Arctic Grayling. *Water resources bulletin 25*: 625–635.

Rivier, B.; Seguier, J. 1985: Physical and biological effects of gravel extraction in river beds. *In*: Alabaster, J.S. *ed*. Habitat modification and freshwater fisheries. London, Butterworths, pp. 131–146.

Rosenberg, D. M.; Wiens, A. P. 1975: Experimental sediment addition studies on the Harris River, N.W.T., Canada: the effect on macroinvertebrate drift. *Verhandlungen, internationale Vereinigung für theoretische und angewandte Limnologie 19*: 1568–1574.

Ryder, G. I. 1989: Experimental studies on the effects of fine sediment on lotic invertebrates. Unpublished PhD thesis. Department of Zoology, University of Otago, Dunedin, New Zealand. 216 p.

Sagar, P. M. 1983: Invertebrate recolonisation of previously dry channels in the Rakaia River. *New Zealand journal of marine and freshwater research 17*: 377–386.

———1986: The effects of floods on the invertebrate fauna of a large unstable, braided river. *New Zealand journal of marine and freshwater research 20*: 37–46.

Scrimgeour, G. J.; Davidson, R. J.; Davidson, J. M. 1988: Recovery of benthic macro-invertebrate and epilithic communities following a large flood, in an unstable, braided, New Zealand river. *New Zealand journal of marine and freshwater research 22*: 337–344.

Scrimgeour, G.J.; Winterbourn, M.J. 1989: Effects of floods on epilithon and benthic macroinvertebrate populations in an unstable New Zealand river. *Hydrobiologia 171*: 33–44.

Sigler, J. W.; Bjorn, T. C.; Everest, F. H. 1984: Effects of chronic turbidity on density and growth of steelheads and coho salmon. *Transactions of the American Fisheries Society 113*: 142–150.

Smith, D. G.; Cragg, A. M.; Croker, G. F. in press: Water clarity criteria for bathing waters based on user perception. *Journal of environmental management*.

Sorensen, D. L.; McCarthy, M. M.; Middlebrooks, E. J.; Porcella, D. B. 1977: Suspended and dissolved solids effects on freshwater biota: a review. United States Environmental Protection Agency. 600/3-77-042.

Soroka, I. K.; McKenzie-Grieve, G. 1983: A biological and water quality assessment at a placer mine on Little Gold Creek, Yukon Territory. Environment Canada, Environmental Protection Service, Pacific Region, Regional Program Report 83-06, Yukon Territory.

Thomas, V. G. 1985: Experimentally determined impacts of a small, suction gold dredge on a Montana stream. *North American journal of fisheries management 5*: 480–488.

Tsui, P. T. P.; McCart, P. J. 1981: Effect of stream-crossing by a pipeline on the benthic macro-invertebrate communities of a small mountain stream. *Hydrobiologia 79*: 271–276.

Van Nieuwenhuyse, E. E.; LaPerriere, J. D. 1986: Effects of placer gold mining on primary production in subarctic streams of Alaska. *Water resources bulletin 22*: 91–99.

Wagener, S. M.; LaPerriere, J. D. 1985: Effects of placer mining on the invertebrate communities of interior Alaska streams. *Freshwater invertebrate biology 4*: 208–214.

Weber, P. K.; Post, R. 1985: Aquatic habitat assessments in mined and unmined portions of the Birch Creek watershed. *Technical report 85-2*. Alaska Department of Fish and Game, Juneau, Alaska. 65 p.

West Coast Goldminers Association. 1988: Treatability study: alluvial gold mining effluent. West Coast Goldminers Association and New Zealand Department of Trade and Industry report. 45 p.

Winterbourn, M. J. 1986. Forestry practices and stream communities with particular reference to New Zealand. *In*: Campbell, I.C. *ed*. Stream protection. The management of rivers for instream uses. Victoria, Chisholm Institute of Technology, Water Studies Centre, pp. 57–74.

————1987: Invertebrate communities. *In*: Viner, A.B. *ed*. Inland waters of New Zealand. Wellington. Science Information Publishing Centre, Department of Scientific and Industrial Research, pp. 167–190.

Winterbourn, M. J.; Cowie, B.; Rounick, J. S. 1984: Food resources and ingestion patterns of insects along a West Coast, South Island, river system. *New Zealand journal of marine and freshwater research 18*: 379–388.

Winterbourn, M. J.; Rounick, J. S. 1985: Benthic faunas and food resources of insects in small New Zealand streams subjected to different forestry practices. *Verhandlungen, internationale Vereinigung für theoretische und angewandte Limnologie 22*: 2148–2152.

Yukon Fisheries Protection Authorization. 1988: Applicable to placer mines in the Yukon Territory. Ottawa, Canada. 21 p.